

Analytical results for sediment samples were discussed in Sections 4.3.2 and 5.0 of the Revised Supplemental Investigation report and are summarized in Table 2-18. As discussed, on-site ditch sediment samples contained similar concentrations of inorganics as off-site ditch sediment samples with the exception of barium. Barium concentrations in on-site sediment samples (SD-1 and SD-2) were 3,590 ppm and 3,220 ppm, respectively; barium concentrations in off-site sediment samples (SD-3 and SD-4) were 121 ppm and 846 ppm, respectively. Cd concentrations in on-site ditch sediment samples were also slightly elevated compared to off-site sediment samples (51.8 ppm and 35.7 ppm, compared to BMDL and 32.5 ppm, respectively). As was discussed in detail in Section 2.2.2.2, the highest erosion losses for Ba and Cd per acre basis was predicted for Area B, which is located adjacent to the drainage ditch from which sediment samples SD-1 and SD-2 were collected. Therefore, it is likely that the concentrations of Ba in Cd measured in these samples are due to erosion of surficial soils from Area B.

In comparing off-site ditch water sample concentrations (SW-3 and SW-4) with off-site ditch sediment sample concentrations (SD-3 and SD-4), it is evident that in most cases inorganics are sorbed onto sediments rather than dissolved to surface water (i.e., concentrations of inorganics are higher in sediment samples than in corresponding surface water samples).

As discussed in Section 4.4.1 and 4.4.4 of the Revised RFI report, water and sediment samples were collected in the DS Tributary by USEPA during the Fields Brook Remedial Investigation (see Tables 4-7 through 4-9 of the Revised RFI report). One sediment sample was collected at the point at which the DS Tributary leaves the RMI property (Station 214), located approximately in the same area as sample DW-G. Sediment was also collected downstream of this location just upstream of State Road (Station 213); further downstream of this location, just downstream of State Road, both water and sediment were collected in the DS Tributary (Station 024).

Table 2-19 presents a comparison of the results of analyses of water from sample DW-G taken during the RFI and USEPA Sample 024 taken in the DS Tributary downstream of the RMI Plant during the Fields Brook Remedial Investigation (USEPA, 1985a). Although it is recognized that these two samples were not collected in the same locations, and were collected several years apart, the table is

TABLE 2-19

COMPARISON OF ANALYSES OF WATER FROM
DW-G AND USEPA SAMPLE (STATION) 024 IN
THE DS TRIBUTARY

Constituent	DW-G ^a (ppb)	USEPA DS Tributary Station 024 ^b (ppb)
As	1.8	No data
Ba	BMDL	650
Cd	2.1	6.2 ^c
Cr	BMDL	BMDL
Pb	3.6	BMDL
Hg	BMDL	0.7 ^c
Ni	BMDL	BMDL
Se	BMDL	No data
Ag	BMDL	BMDL
Zn	77	BMDL ^c

^aSee Table 6-9 of the Revised RFI report and Table 2-17.

^bSee Table 4-7 of the Revised RFI report.

^cData considered questionable by USEPA.

given to show a relative comparison of water quality in the DS Tributary over time. Barium (650 ppb), Cd (6.2 ppb), and Hg (0.7 ppb) were the only metals of interest detected in USEPA sample (station) 024. However, no data are available for As and Se, and results are questionable for several other parameters. Arsenic, Pb, and Zn were detected in DW-G but not in USEPA sample (station) 024. Perhaps the only conclusion to be drawn by a comparison of these two samples is that Ba was detected in fairly high concentrations in USEPA sample (station) 024 and Zn was detected in fairly high concentrations in DW-G, but there is no replication of these results in the two samples.

Table 2-20 is a summary of the sediment data collected by USEPA in the DS Tributary during the Fields Brook Remedial Investigation (USEPA, 1985a). These data represent the highest concentrations measured at various depths at the three locations sampled by USEPA from nearest the RMI facility (Station 214) to furthest downstream station 024 (see Table 4-9 of the Revised RFI report). At Station 214 (located in the same vicinity as DW-G) samples were collected at three depths, 0 to 6 inches; 6 to 12 inches; and 12 to 20 inches. At Station 213, samples were collected at the 0 to 6 inches and 6 to 12 inches depths only; and at Station 024, one depth, 0 to 6 inches, was collected and analyzed in triplicate. In Table 4-9 of the Revised RFI report which shows each of the individual sample results, it appears that in general, the highest concentrations of the constituents of interest were found in the 6 to 12 inches sample for Station 214, and in the 0 to 6 inches sample for Station 213. This distribution of concentrations suggests that perhaps the highest concentrations of constituents are quickly settling out and being accumulated nearest the RMI plant, but that the depth of chemical presence diminishes in the DS Tributary sediments as distance from the RMI plant increases. This trend is also supported by the highest concentrations of the three sampling locations summarized in Table 2-20. As shown in Table 2-20, except for Cr measured at Station 024, the highest concentrations of the constituents of interest in all USEPA samples from each of the three locations were found for Station 214, the station nearest the RMI facility.

2.3.2 Potential for Release and Migration

The potential release of site constituents from on-site groundwater and soils to on-site surface water ditches has been briefly discussed in Sections 2.1 and 2.2. It

TABLE 2-20

**HIGHEST CONCENTRATIONS MEASURED IN DS TRIBUTARY
SEDIMENTS BY USEPA FOR FIELDS BROOK REMEDIAL INVESTIGATION^a**

Constituent	USEPA Station 214 (0-20 inches) ^b	USEPA Station 213 (0-12 inches) ^c	USEPA Station 024 (0-6 inches) ^d
As	21.5	8.8	9.0
Ba	4,139	113.1	2,654
Cd	20.2	0.5	3.01
Cr	36.9	22.8	61.7
Pb	126.3	7.6	Not Analyzed
Hg	1.5	0.4	Not Analyzed
Ni	122.5	27.5	107.5
Se	BMDL	BMDL	BMDL
Ag	BMDL	BMDL	BMDL
Zn	314	113.6	164

^aSee Table 4-9 of the Revised RFI report.

^bMeasurements made for 0 to 6 inches, 6 to 12 inches, and 12 to 20 inches samples. Highest concentrations of these depths reported.

^cMeasurements made for 0 to 6 inches and 6 to 12 inches samples. Highest Concentrations of these depths reported.

^dMeasurements at one depth, 0 to 6 inches, collected in triplicate. Highest concentrations of triplicate analyses reported.

has been shown that water in the wastewater treatment ponds may be recharging the shallow groundwater near Areas D and G and that surface water in the drainage ditch along the eastern property line locally recharges the shallow groundwater. These are the only migration sources from surface water to other on-site media that is considered likely.

It is believed that the concentrations of constituents in the site surface water ditches are a result of contributions from three possible sources: partial discharge of shallow groundwater into the ditches intercepting a portion of the water table zone; erosion of surficial soils; and off-site sources. Leaching of surficial soils to on-site drainage ditches was not determined to be a significant or likely release mechanism (see Section 2.2.2.2). Because water in the site drainage ditches may move off site via the DS Tributary, and because surface water concentrations represent the culmination of constituents from several on-site sources, the surface water pathway is considered the primary pathway of concern at the RMI Sodium Plant. In the sections that follow, the potential for off-site migration of site constituents and the significance of the concentrations found in site surface waters will be evaluated.

2.3.2.1 Potential Groundwater Recharge from Ponds and Subsequent Discharge to On-Site Ditches. As discussed earlier in Section 2.2.2.2, water from the wastewater treatment ponds appears to be recharging the shallow groundwater beneath the site, and thus may be a partial explanation for the source of concentrations of constituents detected in site groundwater samples. Groundwater may then be discharging to some on-site surface water ditches, offering a partial explanation for constituents detected in on-site surface water (ditch) samples.

A relative comparison of constituent concentrations in water and sediment samples collected from the wastewater ponds was made to determine which constituents are more closely associated with water and less strongly sorbed to sediments in the ponds, and thus, are possibly more available to recharge to shallow groundwater on site. This comparison was accomplished by calculating the ratios of sediment to water concentrations for each constituent, from each wastewater treatment pond, relative to the ratios for constituent detection limits. For example, the sediment detection limit for As is 5,000 ppb, and the water detection limit for As is 1.0 ppb; therefore the sediment/water detection limit ratio is 5,000 ppb/1.0 ppb or 5,000. The assumption inherent in this comparison is that if the ratio calculated from the

sediment and water concentrations for a constituent is less than the detection limit ratio, the constituent is more closely associated with the water in the ponds than the sediments, and is therefore more likely to be available to recharge groundwater. If the calculated ratio is greater than the detection limit ratio, the constituent is probably more closely associated with sediments and is more strongly sorbed; and if the calculated ratio equals the detection limit ratio, concentrations were BMDL in both media. An inference can be made about the mobility of the constituents relative to one another based on the frequency of times the calculated ratios were less than or greater than the detection limit ratios, and by how much the calculated ratios deviate from the detection limit ratios.

Table 2-21 presents the results of this comparison of sediment/water ratios for constituents measured in the wastewater treatment ponds. As shown, on the basis of this comparison, the constituents more associated with pond water than sediments (based on the frequency of ratios less than detection limit ratio) are Cd and Hg.

The constituents more closely associated with sediments are (in order): Ba and Pb; Ag; and Cr. Those constituents which were mainly BMDL in both media were As and Se. Therefore, it may be inferred from this comparison that the relative order of mobility of constituents present in the wastewater ponds (in increasing order) is:

$$\text{Ba} \approx \text{Pb} < \text{Ag} < \text{Cr} < \text{Cd} < \text{Hg} < (\text{Se}; \text{As})$$

Although this is a relative comparison, it is remarkably similar to the order of sorption potential predicted by literature-derived K_d values presented in Table 2-10 and discussed in Section 2.2.2.

As stated earlier, this comparison assumes that recharge of shallow groundwater from the wastewater treatment ponds is likely to occur. As discussed in Section 2.2.2.2, because a gradient of surficial to subsurface soil concentrations was not evident in most areas, and because EP Toxicity tests performed on subsurface soils did not indicate the potential for subsurface soils to leach to a significant degree, it was proposed that the elevated concentrations in groundwater could be explained by possible recharge of shallow groundwater by the wastewater ponds. The fact that Cd was detected in french drain samples (see Table 6-8 of the Revised

TABLE 2-21

**RELATIVE COMPARISON OF SEDIMENT/WATER RATIOS
FROM WASTEWATER TREATMENT POND SAMPLES^a**

Pond Sample	Ratio of Sediment to Water Concentrations (ppb/ppb)							Ag
	As	Ba	Cd	Cr	Pb	Hg	Se	
Detection Limits	5,000	40	1,000	500	500	500	500	50
Pond 1	5,000	1,471	294 ^b	398 ^b	4,600	48 ^b	500	220
Pond 2	5,000	41	233 ^b	102 ^b	1,450	15 ^b	500	45 ^b
Pond 3	5,000	258	88 ^b	1,520	2,393	111 ^b	500	180
Pond 4	3,333 ^b	1,589	500 ^b	1,103	1,450	143 ^b	500	230
Pond 5	5,000	1,563	90 ^b	575	1,900	500	500	140

^aConstituents in both media in ppb. Values measured as BMDL were assumed to be equal to the detection limit for purposes of comparison. See Tables 6-6 and 6-7 of the Revised RFI report for raw data.

^bRatio is less than the detection limit ratio. Assumption is: if calculated ratio is less than the detection limit ratio, the constituent is more closely associated with water than sediments; if calculated ratio is greater than detection limit ratio, then more closely associated with sediments; if equal to detection limit ratio, concentrations were BMDL in both media.

RFI report) and was estimated to be fairly mobile in groundwater on the basis of calculated retardation factors (see Table 2-11) supports this assumption. The only inconsistency with this predicted distribution of constituents available for groundwater recharge and the actual groundwater concentrations measured is that Ba was found in relatively high concentrations in groundwater, but was predicted to be strongly sorbed to sediments. This may be partially explained by the following: the concentrations of Ba in pond water are sufficiently high, and the presence of Ba in subsurface soils in elevated concentrations in the area of the wastewater ponds from fill activities (Areas D and G) has been indicated; therefore the existing Ba concentrations in soil and groundwater may exceed the capacity of the soils' continued ability to sorb Ba to a significant degree.

The presence of elevated levels of Ba and Cd in groundwater, whatever its source, may be expected to contribute to concentrations of constituents in some on-site drainage ditches due to discharge of shallow groundwater. However, as discussed in Section 2.1, because the on-site drainage ditches are shallow and do not intercept the entire portion of the water table zone, contributions of constituents from groundwater to on-site surface water ditches are expected to be minimal. This is substantiated by the slow seepage velocities of the constituents calculated using retardation factors (see Table 2-11) and the low concentrations of constituents detected in the on-site ditch water samples (see Table 2-17), especially Ba, relative to groundwater concentrations. Therefore, it appears that the concentrations of constituents detected in the on-site surface water ditches are due to another source, most likely the erosion of surficial soils to the on-site ditches (discussed in Section 2.2.2.2) and the subsequent dynamic interactions with ditch sediments in attaining equilibrium concentrations with water in the ditches.

2.3.2.2 Potential Groundwater Recharge from Eastern Drainage Ditch. As discussed previously in Section 2.1.2, surface water in the drainage ditch located off site along the eastern property boundary of the RMI Sodium Plant site locally recharges shallow groundwater. However, as discussed in Section 2.3.1.2, concentrations of inorganics measured in off-site surface water samples collected from the eastern drainage ditch are relatively low and are believed to be indicative of the water quality of the Ashco Reservoir. Therefore, although groundwater recharge from the eastern drainage ditch to groundwater does occur, the low concentrations of inorganics measured in the off-site surface water samples indicate

that the migration of inorganic constituents to shallow groundwater via recharge from the ditch does not occur to a significant degree.

2.3.2.3 Potential Erosion of Surficial Soils and Transfer to Drainage Ditch

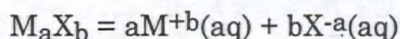
Water. The potential erosion of surficial soils was discussed earlier in Section 2.2.2.2. It was shown that, on a per unit basis, Area F had the highest predicted erosion losses; and on a per area basis, Area B had the greatest predicted erosion losses. As a means of evaluating the site constituents' solubility and therefore potential for mobility in on-site surface water (i.e., the drainage ditches), "theoretical equilibrium concentrations" for each of the constituents were calculated. These theoretical values were then compared to actual values measured in the surface water samples collected from the on-site drainage ditches (samples DW-A through DW-G). This comparison provides a relative estimate of the amount of each constituent being retained by surficial soils, and thereby being unavailable for transport to adjacent surface water. A summary of method by which this comparison was made and the results are described below. Calculations supporting this comparison are given in Appendix A.

In order to compare the "theoretical maximum" or equilibrium concentrations of the metals of concern with the observed values measured in site drainage areas, the solubility products (K_{sp}) for each metal and the possible controlling anions were determined (see Appendix A). This information was then used to calculate equilibrium concentrations of the dissolved metals. K_{sp} values for most commonly occurring compounds can readily be found in chemical handbooks, therefore, if an anion concentration is known, the corresponding metal concentration in equilibrium with the anion can be calculated. This metal concentration will be the maximum concentration in water expected under equilibrium conditions.

For each metal under consideration, the stable valence state in surface water at the pH range measured on site was first determined from Eh-pH diagrams available in the literature. That is, for the oxidizing conditions normally found in shallow surface waters and for the pH range measured (from 6.27 to 7.93, with an average value of 6.91), the stable valence state was determined for each metal. The anions determined to be present under these conditions were CO_3^{2-} , OH^- , and SO_4^{2-} . The concentrations of each of these anions was determined from either the average value measured in the surface water samples taken on site (see Appendix 9 of the

Revised RFI report), or the concentrations were calculated assuming equilibrium with atmospheric carbon dioxide and the pH range measured. Both a high and a low concentration for each anion being considered was used in this calculation, except for SO_4^{2-} , in which an average value was used (see Appendix A). Using the anion concentrations and the calculated solubility products of the various ionic compounds formed (e.g., BaSO_4), the metal concentration in equilibrium with each anion was calculated. As and Se were not included because they form anions in solution, and therefore do not form compounds with the other anions present. The method used to arrive at these calculations is briefly described below.

An ionic compound such as BaSO_4 , will dissociate in water accordingly:



In order to quantify the extent to which a particular compound will dissolve in water, the solubility product (K_{sp}) is used. The solubility product is defined as:

$$K_{\text{sp}} = [\text{M}^{+b}]^a [\text{X}^{-a}]^b$$

where $[\text{M}^{+b}]$ and $[\text{X}^{-a}]$ are the molar concentrations of the metal (cation) and anions respectively. The K_{sp} values are strictly valid for only very low dilute solutions. At higher concentrations it is necessary to use activity, or "effective concentration". Activity is defined as:

$$A_i = [\text{C}_i]\gamma_i$$

where $[\text{C}_i]$ is the concentration of the dissolved species, and γ_i is its activity coefficient.

In very dilute aqueous solutions, activity coefficients (γ_i) are close to unity and ionic strengths (I) approach zero. For more concentrated solutions (up to an ionic strength of about 0.1M), the activity coefficient can be calculated from Debye-Huckel theory:

$$\log \gamma_i = \frac{-A Z_i^2 (I)^{1/2}}{1 + B a_i (I)^{1/2}}$$

A and B are parameters derived from reference tables, and are dependent only on the temperature of the solution. The value, a_o , is also derived from reference tables and is dependent only on the effective size of the hydrated ion of interest (e.g., Ba^{+2}). Z_i is the charge on the ion (e.g., +2 for Ba^{+2}). I is the ionic strength of the solution, and can be calculated from:

$$I = 1/2 \sum [C_i]Z_i^2$$

where $[C_i]$ and Z_i are as previously defined.

Drainage ditch sample DW-B had the highest concentration of dissolved species such as Ca, Mg, Cl, SO_4 , etc. (see Appendix 9 of the Revised RFI report); and an anionic strength (I) of 0.09M was calculated using the formula presented earlier. Most of the drainage ditch samples had ionic strengths much lower than 0.09M, but use of this highest value for I resulted in an activity coefficient (γ_i) for Cr^{+3} of 0.2. Of the ions considered, Cr^{+3} has the activity coefficient farthest from unity (1.0) because it has the highest charge (+3); however, for most of the constituents, γ_i would be closer to unity.

The theoretical maximum or equilibrium concentrations for each metal under consideration and for likely anion species calculated under the assumptions previously described are given in Table 2-22. The theoretical maximum concentrations in water are actually the lowest concentrations formed for each ionic compound of interest. For example, in Table 2-22, $CdCO_3$ has the lowest predicted concentration of all ionic compounds of Cd expected to be formed (1.79 ppb). Therefore, this compound is predicted to drive the solubility of Cd in surface water under equilibrium conditions, i.e., it is the "theoretical maximum". In other words, the lowest concentration for each metal listed in Table 2-22 is the concentration at which that compound would be expected to precipitate, thereby limiting solubility of the metal to the concentration listed. Another example from Table 2-22 is taken for Ba. Under the surface water conditions found in the samples taken, Ba would be expected to precipitate from solution as $BaSO_4$ when the Ba^{+2} concentration reaches 10.6 ppb. Based on the assumptions described, and solubility arguments alone, Ba^{+2} would not be expected to exceed 10.6 ppb in water because precipitation with the sulfate ion would occur at that concentration.

TABLE 2-22

CALCULATED THEORETICAL MAXIMUM CONCENTRATIONS
OF SITE CONSTITUENTS^a

Inorganic Constituents	Expected Valence	CO ₃ ²⁻		OH ⁻		SO ₄ ²⁻
		1.43 x 10 ⁻⁸ M ^b	1.58 x 10 ⁻⁶ M ^c	1.86 x 10 ⁻⁸ M ^b	8.51 x 10 ⁻⁷ M ^c	1.43 x 10 ⁻³ M ^d
Cd	+2	197	1.79 ^e	3.9 x 10 ⁹	1.9 x 10 ⁶	Soluble ^f
Pb	+2	2.18 x 10 ³	19.8 ^e	1.7 x 10 ⁸	8.0 x 10 ⁴	2.61 x 10 ³
Ba	+2	Soluble	7.1 x 10 ⁵	Soluble	Soluble	10.6 ^e
Cu	+2	1.1 x 10 ⁶	1.0 x 10 ⁴	2.9 x 10 ⁴	14.0 ^e	Soluble
Cr	+3	Unstable ^g	Unstable	5.41	5.65 x 10 ^{-5e}	Soluble
Ni	+2	2.71 x 10 ⁷	2.45 x 10 ⁵	4.75 x 10 ⁷	2.27 x 10 ^{4e}	Soluble
Zn	+2	6.86 x 10 ⁴	621 ^e	8.51 x 10 ⁶	4.06 x 10 ³	Soluble
Hg	+2	Unstable	Unstable	1.45 x 10 ⁻²	6.92 x 10 ^{-6e}	Soluble
Ag	+1	2.57 x 10 ⁶	2.44 x 10 ^{5e}	1.16 x 10 ⁸	2.54 x 10 ⁶	Soluble

^aAll concentrations in ppb.

^bAnionic concentration calculated at the lower pH, 6.27 (see Appendix 2).

^cAnionic concentration calculated at the higher pH, 7.93 (see Appendix 2).

^dAnionic concentration calculated for average pH value, 6.91 (see Appendix 2).

^eConcentration which determines the maximum concentration of the dissolved constituent in water.

^fConcentration expected to be higher than highest concentration listed for this constituent.

^gCompound does not precipitate.

Table 2-23 presents a comparison of the maximum concentrations of site constituents measured in on-site surface water ditch samples (see Table 2-17) and the calculated theoretical maximum concentrations (or minimum concentration of all ionic compounds evaluated) for each constituent from Table 2-22. Table 2-23 indicates that of the constituents evaluated, Cd (from DW-B) was the only metal for which the calculated solubility is smaller than the observed metal concentrations in the surface water samples. This supports the view that most of the metals are immobilized in the soil and therefore do not readily reach equilibrium concentrations in the surface water. This is also consistent with the fact that total metal concentrations were measured (rather than dissolved concentrations), so the observed concentrations include dissolved and suspended metals, and are probably higher than the dissolved species concentrations alone. This is especially important in evaluating the Cd concentrations in DW-B. The water in the drainage ditch at location DW-B was at a very low level (approximately 3 inches deep) and it is very likely that some sediment was collected along with the water sample at this location.

It should be noted that this theoretical comparison is dependent upon the assumptions made. For example, only the trivalent state (+3) of Cr was considered in the calculations; some of the Cr may be present as the hexavalent chromate ion. Also, while Ba, Cu, Cr, and Hg were not detected in any of the drainage ditch samples, the detection limits exceed the theoretical maximum values. Although this comparison is theoretical, it is valuable in illustrating the relative predicted solubilities of the site constituents and to what degree they may be expected to be attenuated in surficial soils.

In the previous sections, it has been established that erosion of surficial soils is likely to occur, but that the transfer of dissolved constituents from surficial soils to the drainage ditches via erosion will be limited, based on the predicted solubilities of the constituents evaluated. To further assess the potential contribution of eroded surficial soils to concentrations of constituents on-site ditch water, a comparison was made of the average concentrations of the major waste constituents (Ba, Pb, and Cd) in surficial soil samples nearest the on-site drainage ditches (i.e., those soil samples most representative of what concentrations may potentially erode to the ditches) to the concentrations of Ba, Cd, and Pb in the on-site drainage ditch water

TABLE 2-23

**COMPARISON OF MAXIMUM CONCENTRATIONS OF
SITE CONSTITUENTS IN SURFACE WATER FROM
DRAINAGE DITCHES AND CALCULATED THEORETICAL
MAXIMUM CONCENTRATIONS**

Constituent	Detection Limit in Water (ppb)	Maximum Concentration Detected in Ditch Samples ^a (ppb)	Calculated Theoretical Maximum Concentration ^b (ppb)
As	1.0	1.8 (DW-G)	Not Calculated ^c
Ba	500	BMDL ^d	10.6
Cd	1.0	37.9 (DW-B)	1.79
Cu	20	BMDL	14.0
Cr	5.0	BMDL	5.65 x 10 ⁻⁵
Pb	2.0	4.9 (DW-E)	19.8
Hg	0.4	BMDL	6.92 x 10 ⁻⁶
Ni	100	BMDL	2.27 x 10 ⁴
Ag	1.0	BMDL	2.44 x 10 ⁵
Se	1.0	BMDL	Not Calculated ^c
Zn	10	359 (DW-E)	621

^aHighest value for constituent from data for DW-A through DW-G (see Table 2-17).

Analyses also performed for DW-E and DW-G for Sb, Be, Tl, but all concentrations were BMDL.

^bConcentration which determines the maximum concentration of the dissolved constituent in water. Values from Table 2-21.

^cNot calculated, forms anions in solution.

^dBMDL = Below method detection limit.

samples. The results of this comparison are given in Table 2-24. This comparison is similar to the comparison of wastewater pond sediment/water ratios given in Table 2-21, except that instead of comparing the calculated ratios to detection limit ratios, the results were compared to ratios obtained for ditch sample DW-D, which is considered to be representative of background concentrations in the ditches due to its location (see Figure 2-3). Ditch sample DW-G was not evaluated; because of its location, it would be expected to receive contributions of constituents from all on-site drainage ditches on the south end of the RMI plant.

As shown in Table 2-24, Ba was not detected in any on-site ditch water samples, although it was detected in relatively high concentrations all in nearby surficial soils. This finding supports earlier arguments of the likely sorption of Ba to surficial soils, and the unlikelihood of Ba becoming soluble or being leached from surficial soils. The behavior of Cd in the on-site ditches appears to be different from Ba. Except for location DW-F, Cd was either not detected in either nearby surficial soils or in ditch waters; or it was detected in water only when detected in surficial soils. This suggests that Cd is more soluble than Ba; this has been earlier demonstrated by K_d values, retardation factors (Rd), and solubility calculations. As mentioned in Section 2.3.2.3, the relatively high concentrations of Cd in DW-B is most likely due to erosion of high concentrations of surficial soils in Area B. The behavior of Pb is not as straightforward. In several ditches, including the "background" location (DW-D), Pb was detected in water when not present in elevated concentrations in nearby surficial soils. Also, the highest average concentration of Pb in surficial soils evaluated (335.4 ppm near DW-B) was not associated with detectable Pb concentrations in the nearest ditch water sample (DW-B). The presence of Pb in water collected at DW-D and DW-E, where the drainage ditches originate from off site east of the RMI property may suggest contributions from an off-site source.

2.3.2.4 Potential Contributions from Off-Site Sources. Most of the constituents detected in the various environmental media sampled during the RFI at the RMI Sodium Plant site are explained by contributions via numerous release mechanisms from waste constituents expected to be present in the fill areas and/or the wastewater treatment ponds. However, as mentioned above, the concentrations of Pb detected in ditch samples DW-D and DW-E may suggest an off-site source to the east. Also, Zn concentrations in DW-E and DW-G suggest an off-site source

TABLE 2-24

**COMPARISON OF AVERAGE CONCENTRATIONS OF SURFICIAL SOILS
NEAREST THE DRAINAGE DITCH SAMPLES TO WATER
CONCENTRATIONS MEASURED IN DITCHES**

Drainage Ditch Sample	Nearest Surficial Soil Samples ^a	Average Concentration Nearest Surficial Soils/ Concentration of Water in Ditch (ppm/ppm) ^b		
		Ba	Cd	Pb
DW-Dc ("Background")	SSB-1	124/BMDL ^d = 248	BMDL/BMDL = 1,000	20.6/0.0036 = 5,722
DW-A	SS2-1, SS2-2, SS2-3, SS2-4	595/BMDL = 1,190	7.8/0.0019 = 4,105	80.68/0.0037 = 21,805
DW-B	SS3-1, SS3-2, SS3-3, SS3-4	1,573/BMDL = 3,146	199.3/0.038 = 5,245	355.4/BMDL = 177,700
DW-C	SS1-2, SSB-1	101.3/BMDL = 203	BMDL/BMDL = 1,000	17.8/BMDL = 8,900
DW-E	SSB-9	42.2/BMDL = 84.4	BMDL/BMDL = 1,000	16.6/0.0049 = 3,388
DW-F	SS1-1, SS1-4	66.1/BMDL = 132.2	BMDL/0.0031 = 323	BMDL/0.0038 = 3,947

^aDetermined from expected soil erosion patterns (see Figure 2-3). Data from Appendix 9 of the Revised RFI report.

^bWater concentrations from Table 2-15. DW-G not presented; because of location would be expected to receive contributions from all site drainage ditches on the south end of the RMI property.

^cDitch sample DW-D considered background because located in an area removed from influences of RMI Sodium Plant activities.

^dBMDL = below method detection limits. For water, detection limits are: Ba, 0.5 ppm; Cd, 0.001 ppm; Pb, 0.002 ppm. For soil, detection limits are: Ba, 25.0 ppm; Cd, 1.0 ppm; Pb, 15.0 ppm. Values at BMDL were assumed equal to the detection limits.

contribution (see Section 2.3.1.2). As discussed in detail in Section 6.6 of the Revised RFI report, a variety of organic compounds are strongly believed to originate from off site.

Because of the sorption potentials of the site constituents, elevated concentrations of constituents in drainage ditch sediments may be expected. However, based on USEPA's measurements of concentrations of constituents in sediments collected from the DS Tributary near the RMI Sodium Plant, it appears that concentrations of most constituents in stream sediments diminish considerably with distance from the RMI property. Because of the low velocity of water flowing in the on-site drainage ditches, "wash out" of sediments on site to points downstream is not expected to be significant.

2.3.3 Potential Receptors

As discussed in Section 2.2.3, there are no residential receptors identified in the immediate vicinity of the RMI Sodium Plant. In addition, site access is restricted, thereby potential exposure to on-site surface waters is eliminated to all but RMI-authorized personnel. Therefore, the only potential receptors would be located off site, and the only potential source of exposure via the surface water pathway is from water migrating off site to the southwest in the DS Tributary, and eventually to Fields Brook.

As discussed in Section 4.2.4.2 of the Revised RFI report and Section 2.2.3, there are no federal endangered or threatened species nor federal lands managed for ecological value within a 2-mile radius of the RMI Sodium Plant. There are also no existing or proposed state nature preserves or scenic rivers in that portion of Ashtabula County. Although Walnut Beach Park is within the 2-mile radius of the RMI plant and four threatened plant species have been identified there, there are no conceivable migration pathways of site constituents which could affect these species or the park ecosystem. In addition, there were no farms identified between the DS Tributary and Fields Brook (see Figure 2-1).

As discussed in Section 4.4.1 of the Revised RFI report, designated uses for Fields Brook include limited warm water habitat, agricultural and industrial supply, and primary contact recreation. Fields Brook is not a designated drinking water supply

source. Limited warm water habitats are defined as "waters incapable of supporting reproducing populations of warm water fish and associated vertebrate and invertebrate organisms and plants on a year-round or seasonal basis due to natural conditions, irretrievable man-induced conditions, or the demonstration that meeting the criteria would cause substantial and widespread economic and social impact". The use designation for agricultural supply indicates that the waters are suitable for irrigation and livestock watering without treatment, and the industrial supply designation denotes that the waters are suitable for commercial and industrial uses with or without treatment. "Primary contact recreation" means that, during the recreation season, the waters are suitable for full-body contact recreation such as swimming and canoeing (Ohio EPA, 1987, Ohio Water Quality Standards).

It is doubtful that any of the use designations except industrial supply are actually implemented in Fields Brook at this time. Also, because of the documented organic contamination of the Brook (USEPA, 1985a), it is also doubtful that the waters are capable of supporting any but the most tolerant species of aquatic biota. The Ohio Water Quality Standards do not mention the application of the use designations to tributaries. However, because of the size, depth and flow rates of the DS tributary, it is reasonable to expect that the water use designations realistically only apply at the point where the tributary joins Fields Brook.

The DS Tributary is not of sufficient depth or flow to support many higher species of aquatic biota. Indeed, no aquatic biota were observed in any of the site drainage ditches, including the DS Tributary, during the RFI. However, observations by USEPA representatives in September 1990 noted the presence of frogs in the DS tributary on site. Therefore, higher aquatic species are present periodically in the tributary. However, even though higher aquatic species may be present periodically in the tributary, it still appears that the only potential receptors of constituents potentially emanating from the RMI Sodium Plant site via the DS Tributary would be limited populations of extremely tolerant aquatic plant and animal species present in downstream Fields Brook.

2.3.4 Comparison with Appropriate Criteria

Because the only potential receptors identified in Section 2.3.3 which may come in contact with site surface waters are tolerant aquatic life forms in Fields Brook, a comparison with human exposure limit criteria is not appropriate. Instead, as a conservative comparison, concentrations of site constituents present in drainage ditch sample DW-G, the location deemed most representative of levels of constituents which may be migrating off site because of its location (see Figure 2-3), were compared to Ohio State Water Quality Standards and Federal Ambient Water Quality Criteria (AWQC). Table 2-25 is a summary of this comparison.

As noted in the table, several of the standards/criteria (i.e., for Cd, Cr, Ni, Pb, Ag, and Zn) are dependent upon hardness. Hardness was calculated for the DW-G sample (317 mg/L as CaCO_3) using measured Ca and Mg concentrations (see Appendix 9 of the Revised RFI report) by a procedure recommended in *Standard Methods*, No. 314A (AWWA, 1985). This hardness value was then used to extrapolate to the appropriate Ohio Water Quality Standard using the range of values given in the standards for hardness values of 150 to 500 mg/L CaCO_3 . These extrapolated values are shown in parentheses. In general, as hardness increases, the acceptable water quality criteria also increase.

The hardness-dependent AWQC were calculated based on the measured hardness value for DW-G (317 mg/L) and using the appropriate equations described in the federal guidance (USEPA, 1986b).

As shown in Table 2-25, all of the measured concentrations of constituents in sample DW-G are within both the Ohio Water Quality Standards and the acute and chronic AWQC values, with the exception of Cd for the Ohio Standard for warm water habitat. However, the difference in the measured value at DW-G (2.1 ppb) and the extrapolated Ohio water quality value (1.9 ppb) is considered to be negligible. Also, although warm water habitat criteria may be applied to waters designated as limited warm water habitat, it is not believed that the warm water habitat criteria are appropriate for the DS Tributary (criteria may be varied on a case-by-case basis for the limited warm water habitat use designation (Ohio EPA, 1987)). The Cd value for DW-G is well within the agricultural water supply standard (50 ppb). Therefore, it appears that the concentrations of site constituents

TABLE 2-25

**COMPARISON OF CONCENTRATIONS OF SITE CONSTITUENTS
IN DITCH SAMPLE DW-G WITH EXPOSURE CRITERIA
FOR FRESHWATER AQUATIC BIOTA^a**

Constituent	Concentration in Ditch Sample DW-G ^a	Ohio Water Quality Standards (ppb) ^b		Federal Freshwater Ambient Water Quality Criteria (ppb)	
		Warmwater Habitat ^c	Agricultural Water Supply	Criteria (ppb)	
				Acute	Chronic
As	1.8	190	100	NS ^d	NS
Ba	BMDL	NS	NS	NS	NS
Cd	2.1	0.8-3.1 (1.9) ^e	50	14.4 ^f	2.81 ^f
Cr	BMDL	(+3): 44-118 (81) ^e	100	(+3): 4,471 ^f	533 ^f
		(+6): 10	(total)	(+6): 16	11
Pb	3.6	30	5,000	355 ^f	13.8 ^f
Hg	BMDL	0.2	10	2.4	0.012
Ni	BMDL	167-506 (335) ^e	200	4,436 ^f	230 ^f
Se	BMDL	34	50	260	35
Ag	BMDL	1.3	NS	29.6 ^f	0.12 ^f
Zn	77	99-270 (187) ^e	25,000	120 ^g	110 ^g

^aDetermined to be the surface water sample most representative of levels of constituents which may potentially migrate off site. Hardness calculated from Ca and Mg concentrations; estimated at 317 mg/L as CaCO₃ (see Table 2-17).

^bBased on use designations for Fields Brook and the Ashtabula River (Ohio EPA, 1987, Ohio State Water Quality Standards, Section 3745-1-07). See also Appendix 6 of the Revised RFI report.

^cValues given are for 30 day average.

^dNS = No Standard.

^eValue varies depending upon hardness, ranging from 150 to 500 mg/L as CaCO₃. Approximated value in parentheses based on calculated hardness.

^fCalculated criteria based on hardness (317 mg/L) and appropriate equations in *Quality Criteria for Water* (USEPA, 1986b).

^gValue from Table 8-8 of Vol. I of RFI guidance (USEPA, 1989a). Hardness value of 100 mg/L assumed; therefore criteria would be expected to be substantially higher using measured hardness value (317 mg/L).

present in the surface water drainage ditches at the RMI Sodium Plant which may potentially migrate downstream via the DS Tributary do not present a concern to potential human or environmental receptors. However, the USEPA has determined that a surface water action level has been exceeded for sampling location DW-B for Cd, applying the use designation for the Fields Brook tributary to the on-site tributary. In addition, as stated above in Section 2.3.3, USEPA representatives noted the presence of frogs in the DS tributary on site in September 1990. Therefore, higher aquatic species are present periodically in the tributary.

2.3.5 Assessment of Potential for Exposure

As mentioned throughout portions of Section 2.0, there are several potential on-site sources and a variety of possible release mechanisms which may explain the presence of site constituents in site surface waters. The site constituents detected in pond waters and sediments are believed to be related to plant processes which discharge to the ponds. The presence of constituents in water in the on-site drainage ditches is believed to be primarily due to erosion of elevated levels of surficial soils into the ditches, but there may be other minor contributions from groundwater discharge to the ditches; or, least likely, from leaching of surficial soils and subsequent runoff to the ditches.

Although there are several on-site release mechanisms which may explain the presence of constituents in the site surface waters as discussed above, there is believed to be only one potentially significant source of release of constituents from surface water: the migration of constituents in water via the DS Tributary from the southwest corner of the RMI property (near DW-G). Potential receptors of the site constituents from this migration pathway were determined to be limited populations of extremely tolerant aquatic plant and animal species present downstream in Fields Brook. No human receptors were identified. The DS Tributary is not believed to be of sufficient depth or flow to support higher species of biota even though USEPA representatives noted the presence of frogs in the DS Tributary on the RMI property. Therefore, the likelihood of adverse impacts to aquatic biota is expected to be minimal.

In comparing the concentrations of constituents thought to be representative of potential constituents migrating off site (concentrations at DW-G), it was shown

that none of the water concentrations measured in DW-G exceeded the Ohio Water Quality Standards for Fields Brook and the Ashtabula River for warm water habitats, nor the federal freshwater acute or chronic AWQC, with the exception of Cd (at 2.1 ppb). The Cd concentration at DW-G barely exceeded the Ohio Water Quality Standard of 1.9 ppb. However, the state standards were based on the use designation of "warm water habitat" for Fields Brook and the Ashtabula River which is not considered to be an appropriate designation for the DS Tributary. The Cd concentration in DW-G was, however, well within the Ohio agricultural use criterion of 50 ppb. Therefore, although it may be possible for site constituents to migrate off site via the DS Tributary, the only potential receptors are tolerant aquatic life forms, and the concentrations are expected to be very low, and within state and federal environmental exposure criteria. However, the USEPA has determined that a surface water action level has been exceeded for sampling location DW-B for Cd, applying the use designation for the Fields Brook tributary to the on-site tributary.

2.4 AIR AND SUBSURFACE GAS PATHWAYS

The only possible mechanisms of release for these pathways are volatilization, and fugitive dust emission by erosion of surficial soils into the air. The site constituents are not volatile, therefore volatilization is not a relevant pathway for consideration. This has been confirmed by monitoring for total organic vapors and gases using the HNU photoionizer during various sampling events (see Section 6.4 and 6.6.5 of the Revised RFI report). Therefore, this pathway will not be considered further. It is possible that inorganic site constituents sorbed onto surficial soils could be released to the air as fugitive dust emissions. However, any emissions would be expected to be highly localized (i.e., would not be carried off site), and because there are no off-site receptors in the immediate vicinity of the RMI facility (see Section 2.2.3) and site access is restricted, potential off-site exposure to human or environmental receptors via this pathway is not considered complete, and as such will not be further evaluated in this report. In addition, no sources or potential release mechanisms were considered relevant to this pathway because of the lack of on-site receptors, with the possible exception of RMI workers. However, potential exposures to site workers are regulated by OSHA, and are not relevant to the RFI or CMS process.

2.5 SUMMARY

As described in this section of the RFI report, constituents present in the environmental media on the RMI site are interrelated through a variety of potential release mechanisms and migration pathways. The findings and explanations for the presence of site constituents in the media sampled at the RMI site, as well as the significance of these concentrations with regard to potential receptors, are briefly described in the following paragraphs.

2.5.1 Groundwater

Elevated concentrations of Ba and Cd in shallow groundwater have been detected on site, particularly in the areas north (Area G) and east of the wastewater treatment ponds (Area D). The highest concentration of Ba detected in groundwater was 1900 ppb, in well 8-S near Area G; the highest concentration of Cd was 25.7 ppb, near Area D. The presence of these constituents in groundwater is believed to be due, in part, to recharge of the groundwater from the wastewater treatment ponds, and less likely, from the leaching of subsurface soils or buried wastes. Although recharge of shallow groundwater from the off-site drainage ditch located east of wastewater treatment ponds does occur, only low levels of inorganics have been detected in the ditch water which is believed to originate from the Ashco Reservoir. Therefore, the eastern ditch contributes an insignificant portion of constituents detected in shallow groundwater.

The only potential groundwater migration pathways identified were: discharge of shallow groundwater to deep groundwater; and discharge to on-site surface water ditches. Discharge to deeper groundwater is considered to be highly unlikely due to the low hydraulic conductivity, the thickness of the unweathered glacial till zone, and a minimal hydraulic connection between the shallow and bedrock groundwater due to the relatively high bedrock piezometric surface. In fact, a net upward vertical gradient between the two zones occurs in the vicinity of the wastewater treatment ponds. Discharge of shallow groundwater to site drainage ditches was determined to be a possible migration pathway. However, because the drainage ditches are shallow and do not intercept the entire water table zone, contributions of constituents from groundwater to on-site surface water ditches are expected to be minimal. In addition, because of the low hydraulic conductivity of the water table

zone and the predicted attenuation of site constituents based on-site soil properties and the physical/chemical properties of the constituents, site constituents in groundwater are expected to migrate off site at a slow rate. As presented in Table 2-11, retardation factors calculated for the site constituents indicated that the constituents are being effectively attenuated in subsurface soils, and are moving much more slowly than the mass flow of groundwater.

Groundwater in the uppermost water-bearing zone beneath the RMI Sodium Plant is characterized by low hydraulic conductivity and, subsequently, low yield. Therefore, groundwater in this water bearing zone is not expected to serve as a drinking water source. In addition, no receptors of shallow groundwater in the vicinity of the RMI plant were identified because there are no private, domestic, or municipal drinking water wells screened in the shallow groundwater zone or springs used as a source of drinking water within a 5 km radius of the site. In addition, because of low groundwater yields, the majority of the local population relies on the abundant surface water supplies available for drinking water. Because there were no receptors, comparisons with exposure criteria were not performed. Therefore, no exposure to site constituents via the groundwater pathway was predicted.

2.5.2 Soil

Both surficial and subsurface soils were collected at various locations around the RMI plant site. Compared to background concentrations, Ba, Cd, Pb, Ni and As in Areas B and C (combined); Ba, Pb, Ni, and As in Area F; and Ba, Cd, Ni, and As in Area G were determined to be present in surficial soils at elevated concentrations. Areas B and C combined had the highest average surficial soil concentrations for all site constituents except Cr and Se. The subsurface soils in elevated concentrations were determined to be: Area D, up to between 6.5 and 13.3 feet for Ba, Pb, and Ni; and Area G for a variety of constituents at several different depths. The highest overall subsurface soil concentrations were found in Area G, for all constituents except As and Se. The presence of these constituents in these areas was consistent with what was known about the placement of wastes in the Areas D and G. The concentrations of constituents in surficial soils generally appeared to be greatly attenuated with depth. However, in Area G, no "gradient" of waste constituents was observed which indicated that the distribution concentration of constituents

with depth was representative of the placement of wastes over time, rather than the downward leaching of constituents. Arsenic was found in remarkably consistent concentrations throughout the site. It is thought that the levels found were either natural to the regional area, or were the result of agricultural use of arsenic-containing pesticides on the soil before soil fill was imported to the site.

The most likely and significant soil migration pathway was determined to be the erosion of surficial soils to on-site surface water ditches. Predicted erosion losses via precipitation from the waste management areas were quantified by use of the USLE. On a per unit basis, Area F had the highest estimated overall soil losses, with Ba as the highest constituent lost, at a rate of 2.19 lb/yr. On a per acre basis, however, Area B was predicted to have the highest overall losses, again with Ba as the highest rate of loss, at 5.43 lb/yr. Other potential migration pathways were considered possible, such as the leaching of surficial soils to shallow groundwater and/or to surface water; and the leaching of subsurface soils to shallow groundwater. However, these pathways are not expected to be significant because of the likelihood of a high degree of sorption of the site constituents to site soils, based on the chemical and physical properties of the constituents and of the soils (e.g., K_d values, CEC, organic content). In addition, EP Toxicity tests performed on subsurface soils with the highest concentrations of Cd and Pb indicated that leaching of subsurface soils is not likely to occur to a significant degree. Cd was predicted to be one of the most mobile of the site constituents.

Because access to the site is restricted, and there were no receptors identified in the immediate vicinity of the RMI plant, comparison to exposure criteria was not considered appropriate. Instead, the predicted erosion losses were compared the proposed municipal sewage sludge disposal loading rates for the site constituents. All predicted erosion rates were far below federally proposed sewage sludge disposal loading rates. Using the most conservative values of erosion loss, it appeared that none of the constituent concentrations in surficial soils in any waste management areas were at potential levels of concern, with regard to erosion.

2.5.3 Surface Water

Samples were collected from the wastewater treatment ponds, the french drain system, and the site drainage ditches. Barium and Cd were found in all of the

ponds, with Ba in the highest concentrations in both the pond water (at 5,500 ppb in Pond 3) and pond sediments (3,020 ppm in Pond 4). The concentrations of constituents in the french drain samples were substantially lower than the pond water samples, with Cd at 26.8 ppb being the highest constituent level detected. Very low concentrations were found for most constituents in the on-site ditch samples; the highest levels detected were: Zn at 359 ppb at DW-E (and at 77 ppb at DW-G) and Cd, at 37.9 ppb at location DW-B. Because of the location of DW-E (of the southeast corner of the property, where the ditch originates from off site), it was speculated that the Zn could be attributed to an off-site source to the east. The relatively high concentration of Cd at location DW-B is believed to most likely be the result of elevated concentrations of eroded surficial soils originating from Area B. Surface water samples collected from the off-site ditch located to the east of the wastewater treatment ponds during the Supplemental Investigation are believed to be indicative of the water quality from the Ashco Reservoir. Sediment samples collected in the vicinity of DW-B during the Supplemental Investigation (SD-1 and SD-2), indicated the Ba concentrations were elevated and Cd concentrations were slightly elevated where compared to off-site sediment sample SD-3 and SD-4. These sediment concentrations are also believed to be the result of erosion of surficial soils from Area B.

There were several potential release mechanisms identified which may explain the presence of site constituents in the drainage ditches, including erosion of surficial soils; discharge of shallow groundwater to ditches; and possibly, leaching of surficial soils and subsequent transport to ditches via runoff. However, as discussed previously, only the potential erosion of surficial soils to the surface water ditches was determined to be of significance. Calculations of theoretical maximum concentrations expected in water based on solubilities of the constituents indicated that sorption of the constituents was likely to be rather significant, and would limit the expected concentrations of constituents in ditch water. The only exception was Cd, which was predicted at lower concentrations than actually measured (at DW-B). This may have been due to the presence of suspended particulates in the water sample.

In comparing the concentrations of the principal waste constituents (Ba, Cd, and Pb) measured in on-site ditch water samples with nearby surficial soil concentrations, it was shown that Ba appeared to be strongly sorbed, as it was

present in relatively high concentrations in soil, but was never detected in the on-site ditch water samples collected. Cadmium was usually detected in water at low concentrations when it was present in soil; and the behavior of Pb was inconclusive.

No human receptors of surface water potentially migrating off site were identified, and the only likely environmental receptors were determined to be extremely tolerant aquatic species possibly present in downstream Fields Brook. The DS Tributary was not believed to be capable of supporting fish or higher forms of aquatic species. The concentrations of constituents in water from location DW-G (considered to be representative of what may be migrating off site) were compared to the Ohio Water Quality Standards and to federal AWQC. The concentrations in DW-G were found to be below all criteria for all constituents, except the Ohio Warm Water Habitat Standard for Cd (1.9 ppb). This was not determined to be significant, as it is highly unlikely that the DS Tributary would meet the requirements of a warm water habitat. However, the USEPA has determined that a surface water action level has been exceeded for sampling location DW-B for Cd, applying the use designation for the Fields Brook tributary to the on-site tributary. In addition, in September 1990, USEPA representatives observed frogs in the DS tributary on site. Therefore, higher aquatic species are present periodically in this tributary.

2.5.4 Air

No sources or potential release mechanisms were considered relevant to the air pathway because of the lack of on-site receptors, with the possible exception of RMI workers. However, potential exposures to site workers are regulated by OSHA and are not relevant to the RFI or CMS process. Although no air monitoring data are available for metals, it is possible that trace quantities of metals which may be sorbed to surficial soil may migrate via fugitive dust.

2.6 RISK ASSESSMENT FOR INGESTION OF SHALLOW GROUNDWATER

As discussed in Section 1.2.2.2, shallow groundwater at the site may potentially be impacted by SWMUs at the RMI Sodium Plant as evidenced by somewhat elevated

concentrations of Ba and Cd in shallow groundwater samples from a localized area. However, in almost all cases, concentrations of Ba and Cd measured in the most recent downgradient shallow groundwater samples collected (as well as the concentrations of other inorganics detected) were below current maximum contaminant levels (MCLs) promulgated under the Safe Drinking Water Act (see Table 4-4 of the Revised Supplemental Investigation report).

The bedrock zone has not been impacted by site activities. As discussed in Section 6.1.2 of the Revised RFI report and in Section 4.3.1 of the Revised Supplemental Investigation report, although elevated concentrations of Ba have been detected in some bedrock monitoring wells at the site, these concentrations are believed to be naturally-occurring because high Ba concentrations are known to occur in the bedrock groundwater in the region. In addition, hydraulic conductivity, hydraulic gradient, and major ion data all indicate that Ba in the bedrock groundwater could not have originated from the shallow groundwater zone.

Potential receptors for the groundwater pathway were discussed in Section 2.1.3. As discussed, because of low groundwater yields and abundant surface water supplies (i.e., Lake Erie or reservoirs), there are no private, domestic, or municipal drinking water wells screened in the shallow groundwater unit and no springs are used as a source of drinking water within a 5 km radius of the RMI Sodium Plant site (although some domestic wells are present within a 5 km radius of the site, all are screened in the bedrock zone). Therefore, there are no potential human receptors of shallow groundwater at or in the vicinity of the RMI Sodium Plant. It has also been determined that the shallow groundwater in the vicinity of the RMI Sodium Plant is characterized by low hydraulic conductivity and, subsequently, low yields. Therefore, groundwater in this water-bearing zone is not expected to serve as a source of drinking water.

In spite of the fact that no human receptors have been identified for shallow groundwater in the vicinity of the RMI site, and none are reasonably anticipated due to low yield, a risk assessment for the ingestion of constituents in groundwater was requested by the USEPA Region V on June 13, 1991, and is presented in the following sections. Only data for constituents detected in shallow groundwater samples from monitoring wells on the RMI property are utilized in the risk assessment process because the bedrock zone has not been affected by SWMUs at

the RMI Sodium Plant site. Organics detected in groundwater monitoring wells on the southern boundary of the site were demonstrated to have resulted from off-site sources and were not utilized in the conduct of the risk assessment. The following risk assessment was conducted based upon the framework given in the current federal risk assessment guidance document: *Risk Assessment Guidance for Superfund--Human Health Evaluation Manual, Part A* (USEPA, 1989b) and in USEPA's "Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors" (Supplemental Guidance; USEPA, 1991a). In accordance with these guidance documents, relevant potentially exposed population(s) are identified, intakes for the constituents of interest in shallow groundwater are estimated, and a toxicity assessment for the constituents of interest is conducted. Once intakes have been estimated for the relevant potentially exposed population(s), these values are used in the risk characterization, along with the relevant toxicity values, to estimate potential risks posed by ingestion of shallow groundwater.

2.6.1 Quantification of Groundwater Intake

The quantification of groundwater intake will be conducted in two parts (USEPA, 1989b): the estimation of exposure concentrations in shallow groundwater which are expected to be contacted over the exposure period; and the calculation of "intakes" or normalized exposure estimates which describe the mass of a constituent expected to be in contact with the human body per unit body weight, per unit time (e.g., in units of mg/kg-day).

2.6.1.1 Summary of Measured Concentrations. Locations of site monitoring wells are shown in Figure 1-1. Historical and recent analytical data for samples collected from groundwater monitoring wells at the RMI Sodium Plant site are summarized in Table 4-4 of the Revised Supplemental Investigation report. As shown in Table 4-4, monitoring wells at the site have been sampled on several occasions (November 1988, January 1989, and February/March 1991) and samples have been analyzed for dissolved As, Ba, Cd, Cr, Cu, cyanide, Pb, Hg, Ni, Se, Ag, and Zn. During the most recent sampling period (February/March 1991), samples were analyzed for both total and dissolved metals. As shown in Table 4-4 of the Revised Supplemental Investigation report, total metal concentrations were comparable to or greater than dissolved metal concentrations. At the request of

USEPA, total metal concentrations will be used to conservatively estimate risk associated with shallow groundwater at the RMI Sodium Plant site.

Total metal concentrations for the most recent samples (February/March 1991) collected from seven shallow monitoring wells (3-S, 4-S, 5-S, 6-S, 7-S, 9S, and 10S) at the RMI Sodium Plant site are summarized in Table 2-26. Samples were not collected from shallow monitoring well 8-SR (the replacement well for well 8-S) because the well was dry and had not recharged with groundwater at the time of sampling. Samples were also not collected from wells 1-S and 2-S due the presence of DNAPL migrating onto the property from an off-site source (see Section 2.3 of the Revised Supplemental Investigation report). Data for shallow monitoring wells 12-S and 13-S are not included in Table 2-26 because these wells are located outside of RMI property and isolated from the RMI site by a groundwater divide. In addition, the groundwater quality in well 12-S is believed to be influenced by the coal pile located to the east of the site (see Section 4.3.1 of the Revised Supplemental Investigation report). Therefore, data from wells 12-S and 13-S will not be used when estimating potential risks associated with groundwater at the RMI Sodium plant site.

Data for monitoring wells 9-S and 10-S are also included in Table 2-26. However, as discussed in Section 4.3.1 of the Revised Supplemental Investigation report, these wells are considered to be background wells due to their location with respect to the SWMUs. Because groundwater in these wells is not impacted by the SWMUs (historically, constituents of interest have not been detected or have been measured in only low concentrations in these monitoring wells; see Table 4-4 of the Revised Supplemental Investigation report), as a conservative measure, these data will not be used when estimating risk associated with shallow groundwater at the project site.

A data summary for the downgradient shallow groundwater data discussed above is presented in Table 2-27. Included in this table is the number of samples in which concentrations were measured above detection limits; the total number of wells sampled; the frequency of detection; the arithmetic mean concentration; the minimum and maximum detected values; and the upper 95th percent confidence limit on the arithmetic mean. As further discussed in Section 2.6.1.2, the upper 95th percent confidence limit values will be used when estimating exposure

TABLE 2-26

SUMMARY OF SHALLOW GROUNDWATER DATA

Constituent	Detection Limit (mg/l)	Well 3-S (mg/l)	Well 4-S (mg/l)	Well 5-S (mg/l)	Well 6-S (mg/l)	Well 7-S (mg/l)	^a Well 9-S (mg/l)	^a Well 10-S (mg/l)
TOTAL INORGANICS								
Arsenic	0.005	0.0131	BMDL	BMDL	BMDL	BMDL	BMDL	0.0081
Barium	0.200	1.40	0.560	0.910	2.00	0.270	BMDL	0.250
Cadmium	0.001	BMDL	0.0040	0.0041	0.0079	0.0014	BMDL	BMDL
Chromium	0.002	0.0198	0.0059	0.0085	BMDL	0.0082	0.0027	0.0196
Copper	0.020	0.040	0.030	0.030	BMDL	BMDL	BMDL	0.030
Cyanide	0.020	0.030	BMDL	BMDL	BMDL	BMDL	BMDL	BMDL
Lead	0.003	0.0118	0.0054	0.0114	BMDL	0.0044	0.0036	0.0187
Mercury	0.0002	BMDL	BMDL	BMDL	BMDL	BMDL	BMDL	0.0011
Nickel	0.002	0.0868	0.0206	0.0632	0.0636	0.0177	0.0082	0.032
Selenium	0.005	BMDL	BMDL	BMDL	BMDL	BMDL	BMDL	BMDL
Silver	0.020	BMDL	BMDL	BMDL	BMDL	BMDL	BMDL	BMDL
Zinc	0.015	0.108	0.044	0.064	0.136	0.022	0.024	0.076

^a
Background wells

^b
BMDL = Below Method Detection Limit

TABLE 2-27

SUMMARY OF CONSTITUENTS DETECTED IN DOWNGRAIDENT SHALLOW GROUNDWATER^a

Constituent	Number of Detections ^b	Total Number of Samples	Frequency of Detection (%) ^c	Arithmetic Mean (mg/l) ^d	Minimum Detected Value ^e (mg/l)	Maximum Detected Value ^e (mg/l)	Upper 95% ^f Limit (mg/l)
TOTAL INORGANICS							
Arsenic	1	5	20.0	6.62E-03 ^g	0.0131	0.0131	1.01E-02
Barium	5	5	100.0	1.028	0.270	2.00	1.68
Cadmium	4	5	80.0	3.68E-03	1.40E-03	7.90E-03	6.31E-03
Chromium	4	5	80.0	8.88E-03	5.90E-03	0.0198	1.52E-02
Copper	3	5	60.0	0.0280	0.030	0.040	3.60E-02
Cyanide	1	5	20.0	0.0220	0.030	0.030	2.63E-02
Lead	4	5	80.0	7.20E-03	4.40E-03	0.0118	1.11E-02
Mercury	0	5	0.0	2.00E-04	-- ^h	--	--
Nickel	5	5	100.0	0.0504	0.0177	0.0868	7.91E-02
Selenium	0	5	0.0	0.005	--	--	--
Silver	0	5	0.0	0.0200	--	--	--
Zinc	5	5	100.0	0.0748	0.022	0.136	0.119

^a

Does not include data for background wells 9-S and 10-S or off site wells 12-S and 13-S (see discussion in Section 2.6.1.1).

^b

Does not include concentrations measured below minimum detection limits ("BMDL" data).

^c

Ratio of total number of samples above minimum detection limit/total number of samples.

^d

When analytical result was "BMDL", the detection limit was used to calculate the arithmetic means and upper 95% values.

^e

Does not include "BMDL" data.

^f

Calculated as described in text.

^g

Scientific notation is equivalent to multiplying by 10 to an exponent, e.g., 6.62E-03 is equivalent to 6.62×10^{-3} .

^h

Dashes (--) indicate that constituent was not detected.

concentrations for shallow site groundwater, as recommended in the current federal risk assessment guidance (USEPA, 1989b). Definition and calculation of these parameters are described in the following paragraphs.

The number of samples in which concentrations were measured above detection limits and the total number of wells sampled (5 wells: 3-S, 4-S, 5-S, 6-S, and 7-S) were used to calculate the percent frequency of detection. The number of samples in which concentrations were measured above detection limits did not include data points which were below method detection limits ("BMDL").

In order to calculate the arithmetic mean, the detection limit was assigned to sample results which were "BMDL". Although this method of assigning a value to data which were below detection limits is conservative, it allows for the consistent incorporation of all data in the determination of the arithmetic average for each constituent.

The minimum and maximum detected values represent a range of detectable concentrations for samples collected from the seven wells under consideration (this range does not include data which were "BMDL").

The 95th percent upper confidence limit on the arithmetic mean was also calculated, consistent with the most recent federal risk assessment guidance (USEPA, 1989b). The 95th percent upper confidence limits were calculated using the following formula (Gilbert, 1987):

$$UL_{1-\alpha} = \bar{x} + t_{1-\alpha, n-1} \left(\frac{s}{\sqrt{n}} \right)$$

where:

- UL₁ = Upper one-sided confidence limit
- α = Confidence limit (e.g., equal to 0.05 at the 95 percent confidence level)
- \bar{x} = Arithmetic mean of n measurements
- t = Quantities of the t distribution (from statistics table), e.g., t value at the 95 percent confidence level
- n = Number of measurements in data set
- s = Standard deviation of the arithmetic mean
- n-1 = Degrees of freedom

The arithmetic means used were calculated as discussed above. The variance (s^2) of each sample population was determined by the use of the divisor (n-1) to yield an unbiased estimate of the variance of the true population. The standard deviation (s) as used in the determination of the upper 95th percent confidence limits was calculated as the square root of this unbiased estimator, resulting in a reasonable estimate of the standard deviation of the true population (Gilbert, 1987).

2.6.1.2 Intake Parameters. As previously discussed, because of low groundwater yields and abundant surface water supplies, there are currently no private, domestic, or municipal drinking water wells which withdraw water from the shallow groundwater unit on site or within a 5 km radius of the RMI Sodium Plant site. Therefore, no current receptors of constituents in shallow groundwater on or off site have been identified and risks will not be estimated for current populations.

Because of low groundwater yields in the shallow groundwater unit and abundant surface water supplies, it is also very unlikely that drinking water wells would be installed in the shallow groundwater unit at the RMI Sodium Plant site in the future. In addition, it is planned that deed restrictions will be in effect which will prevent installation of drinking water wells on site as part of the selected corrective measures. Therefore, potential future on-site receptors of constituents in shallow groundwater have not been identified and risks will not be estimated for this exposure scenario.

As was discussed in Section 4.6.2 of the Revised RFI report, the RMI Sodium Plant site is located in a highly industrialized area, and is currently surrounded in all directions by industrial or commercial facilities: the Detrex facility located on the southern boundary; the Elkem Metals complex to the east; Ray Roofing and Occidental Chemical Corporation located along the western boundary; and three coal-fired power plants located to the north and east (see Figure 2-24 of the Revised RFI report). The current zoning map for Ashtabula Township indicates that the RMI Sodium Plant site and areas immediately adjacent to the site are zoned for heavy and light manufacturing. Permissible uses of heavy and light manufacturing districts include development of manufacturing, processing, warehousing, wholesale businesses, and major research and testing operations (Ashtabula Township, 1991). Therefore, development of land in the vicinity of the RMI Sodium Plant site for purposes other than industrial or commercial uses is unlikely and the potential exposure of future off-site residential/recreational populations to constituents in shallow groundwater will not be quantified.

It is also not anticipated that groundwater wells would be installed in the vicinity of the site for use by future occupational populations because of abundant surface water supplies and low groundwater yields. However, for the purposes of fulfilling the Agency's request, the ingestion of shallow groundwater by an off-site future occupational population from wells installed in this area (which will be assumed to be a commercial/industrial land use area) will be entertained as a potential future off-site exposure scenario. Ingestion of shallow groundwater will be the only exposure route considered because it is expected that, compared to ingestion, dermal contact with inorganics in shallow groundwater (e.g., via showering or bathing) would be negligible and that ingestion of shallow groundwater would represent the worst-case exposure scenario.

In accordance with the USEPA's Supplemental Guidance (USEPA, 1991a), occupational scenarios should be evaluated when land use is (or is expected to be) commercial/industrial. Chronic Daily Intakes (CDIs) for the ingestion of shallow groundwater will be estimated by the use of the intake parameters recommended in the Supplemental Guidance for commercial/industrial land use and the intake equation for water ingestion given in the current federal risk assessment guidance (USEPA, 1989b). The water ingestion intake equation is given as:

$$\text{Chronic Daily Intake (CDI, mg/kg-day)} = \frac{\text{CW} \times \text{IR} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

where:

CW = Chemical Concentration in Groundwater (mg/L)

IR = Ingestion Rate (liters/day)

EF = Exposure Frequency (days/year)

ED = Exposure Duration (years)

BW = Body Weight (kg)

AT = Averaging Time (days)

Variables used in the intake equation are briefly described in the paragraphs below. It should be noted that the CDIs calculated with this equation are expressed as the amount of chemical at the exchange boundary of the body (e.g., gastrointestinal tract) and available for absorption for the ingestion pathway.

Exposure Concentration (CW). The concentration term in the intake equation is the concentration of a constituent in a given medium that is contacted over the exposure period. For the estimation of groundwater intakes for the RMI Sodium Plant site, the upper 95th percent confidence limits presented in Table 2-27 will be used, as recommended in the current federal risk assessment guidance (USEPA, 1989b).

Ingestion Rate (IR). In accordance with the Supplemental Guidance (USEPA, 1991a), it is assumed that half of an individual's daily water intake (1 liter out of 2 liters) occurs at work. All water ingested is assumed to come from the potentially impacted drinking water source (i.e., bottled water is not considered).

Exposure Frequency (EF) and Exposure Duration (ED). Exposure frequency and duration are used to estimate the total time of exposure. In accordance with the Supplemental Guidance (USEPA, 1991a), for the occupational scenario, it is assumed that an individual works 5 days/week for 50 weeks/year (i.e., EF is equal to 250 days/year). The individual is also assumed to work 25 years at the same location (i.e., ED is equal to 25 years).

Body Weight (BW). The value for body weight is the average body weight over the exposure period. In accordance with the Supplemental Guidance (USEPA, 1991a), only the adult population is considered for the occupational scenario and body weight is assumed to be 70 kg.

Averaging Time (AT). The averaging time selected depends on the type of toxic affect being assessed. When evaluating long-term exposure to noncarcinogenic toxicants, intakes are calculated by averaging intakes over the period of exposure (i.e., ED). For carcinogens, intakes are calculated by prorating the total cumulative dose over a lifetime of 70 years (i.e., chronic daily intakes, also called lifetime average daily intake). The distinction relates to the most conservative currently held scientific opinion that the mechanism of action for potential toxic effects for carcinogens and noncarcinogens is different. The approach for carcinogens is based on the assumption that a high dose received over a short period of time is equivalent to a corresponding low dose spread over a lifetime (USEPA, 1989b). For noncarcinogens, the AT is 9,125 days (365 days/year multiplied by 25 years (ED)). For carcinogens the AT is 25,550 (365 days/year multiplied by 70 years (ED)).

2.6.1.3 Summary of Intakes. Summaries of the CDIs for ingestion of site constituents in shallow groundwater by the potential future occupational population using the upper 95th percent shallow groundwater concentrations, the intake equation, and exposure assumptions described previously are presented in Table 2-28. Intakes for both valence states for Cr (III and VI) are included because, as further discussed in Section 2.6.2.2, toxicity factors are available for both valence states. The estimated CDI values will be used in conjunction with the relevant toxicity values presented in Section 2.6.2 (Toxicity Assessment) to estimate potential risk.

2.6.2 Toxicity Assessment

The purpose of the toxicity assessment is to weigh available evidence regarding the potential of constituents of interest to cause adverse effects in exposed individuals, and to provide, where possible, an estimate of the relationship between the extent of exposure to a constituent and the increased likelihood and/or severity of adverse effects (USEPA, 1989b).

TABLE 2-28

SUMMARY OF INTAKES FOR INGESTION OF SHALLOW GROUNDWATER

POTENTIAL FUTURE OCCUPATIONAL POPULATION^a

Constituent	Intake Parameters ^b					AT Noncarcinogens (days)	AT Potential Carcinogens (days)	Chronic Daily Intake Potential Carcinogens (mg/kg-day)	Chronic Daily Intake Noncarcinogens (mg/kg-day)
	CW (mg/l)	IR (l/day)	EF (d/yr)	ED (yr)	BW (kg)				
Arsenic	1.01E-02 ^c	1	250	25	70	9125	25550	3.5E-05	9.9E-05
Barium	1.68	1	250	25	70	9125	25550	5.9E-03	1.6E-02
Cadmium	6.31E-03	1	250	25	70	9125	25550	2.2E-05	6.2E-05
Chromium (III)	1.52E-02	1	250	25	70	9125	25550	5.3E-05	1.5E-04
Chromium (VI)	1.52E-02	1	250	25	70	9125	25550	5.3E-05	1.5E-04
Copper	3.60E-02	1	250	25	70	9125	25550	1.3E-04	3.5E-04
Cyanide	2.63E-02	1	250	25	70	9125	25550	9.2E-05	2.6E-04
Lead	1.11E-02	1	250	25	70	9125	25550	3.9E-05	1.1E-04
Mercury	-- ^d	1	250	25	70	9125	25550	--	--
Nickel	7.91E-02	1	250	25	70	9125	25550	2.8E-04	7.7E-04
Selenium	--	1	250	25	70	9125	25550	--	--
Silver	--	1	250	25	70	9125	25550	--	--
Zinc	0.119	1	250	25	70	9125	25550	4.2E-04	1.2E-03

^a

Residential/recreational populations not considered because land use on site and in the vicinity of the site is anticipated to be commercial/industrial in the future.

^b

Per the USEPA Supplemental Guidance (USEPA, 1991). See Section 2.6.1.2 for definition of intake parameters.

^c

Scientific notation is equivalent to multiplying by 10 to an exponent, e.g., 1.01E-02 is equivalent to 1.01×10^{-2} .

^d

Dashes (--) indicate that constituent was not detected.

Toxicity assessments are generally accomplished in two steps: hazard identification and dose-response assessment. Hazard identification is the process of determining whether exposure to a constituent can cause an increase in the incidence of an adverse health effect (e.g., cancer), and whether the effect is likely to occur in humans. The dose-response evaluation is the process of quantitatively evaluating the toxicity information and characterizing the relationship between the dose of the constituent and the adverse health effects in an exposed population. From this quantitative dose-response relationship, toxicity values (discussed below) may be derived and further used to estimate the incidence of adverse effects as a function of potential human exposure to the constituent (USEPA, 1989b). These toxicity values are used later in the risk characterization step of the risk assessment process (see Section 2.6.3) to quantify potential human exposures to site constituents in shallow groundwater.

Although the toxicity assessment is an integral component of the risk assessment process, the amount and type of toxicological information available are limited in most cases (USEPA, 1989b). The USEPA has performed the toxicity assessment step for numerous chemicals and has made available the resulting toxicity information and toxicity values through their on-line toxicity data base, the Integrated Risk Information System (IRIS). IRIS was originally developed to make chemical-specific risk information readily available to members within the USEPA and state agencies involved in risk assessments, and to promote consistency in the performance of risk assessments and subsequent risk management decisions. The information contained in Section I (Chronic Health Hazard Assessment for Noncarcinogenic Effects) and Section II (Carcinogenicity Assessment for Lifetime Exposure) of the IRIS chemical files represents a consensus judgment of USEPA's Reference Dose (RfD) Work Group or Carcinogenic Risk Assessment Verification Endeavor (CRAVE) Work Group, respectively. These two Agency-wide Work Groups include scientists from USEPA's program offices (e.g., hazardous waste, air, pesticides) and the Office of Research and Development. Individual USEPA offices have conducted comprehensive scientific reviews of the literature available on particular chemicals, and have performed the hazard evaluation and dose-response assessment. These assessments have been summarized for IRIS and reviewed and revised by the appropriate Work Group. As new information becomes available, these Work Groups will re-evaluate their work and revise IRIS files accordingly. Because the toxicity information is constantly being updated, IRIS is currently only

available on-line. As of April 1988, the IRIS database was made available to the public, and all USEPA staff, USEPA contractors, and PRPs (or their consultants) are expected to use IRIS as the primary source of toxicity information in performing risk assessments (Federal Register, 1988).

Second to IRIS, the USEPA recommends that the Health Effects Assessment Summary Tables (HEASTs) be consulted. Formerly called "The Quarterly" and associated references, HEASTs are tabular presentations of toxicity information and values for chemicals for which Health Effects Assessments (HEAs), Health and Environmental Effects Documents (HEEDs), Health and Environmental Effects Profiles (HEEPs), Health Assessments Documents (HADs), or Ambient Air Quality Criteria Documents (AAQCDs) have been prepared. The HEASTs summarize interim (pending IRIS verification) reference doses (RfDs) for noncarcinogens and slope factors (SFs) for potential carcinogens, as well as other toxicity information for specific chemicals. In some cases, values which are verified and appear in IRIS are also listed in the HEASTs. Therefore, the HEASTs are especially helpful when verified information for a chemical is pending verification but it not yet available in the IRIS database. The HEASTs are updated and issued once per quarter, but are often not available for many months after the end of a given quarter. The latest available HEAST at this time is the Annual FY-1991, January 1991 (USEPA, 1991b).

Toxicity values are derived separately for potential carcinogens and noncarcinogens. Noncarcinogenic effects, carcinogenic effects, or both types of effects may be associated with a single constituent. The types of toxicity values available for noncarcinogens are the chronic RfDs, formerly called ADIs or AICs. The chronic RfD is an estimate of the daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime, usually in units of mg/kg-day. The greater the value of the RfD, the less toxic the chemical; doses that are less than the RfD are not likely to be associated with adverse health effects. Usually, as the frequency of exposures exceeding the RfD increases, and as the size of the excess increases, the probability increases that adverse health effects may be observed in a human population. RfDs are usually determined from laboratory studies on animals, using a lowest-observed-adverse-effect-level (LOAEL) or a no-observed-adverse-effect-level (NOAEL), divided by appropriate uncertainty factors (UF) and modifying

factors (MF) to account for differences in human and animal sensitivities, etc. Noncarcinogens are usually assumed to have a "threshold", i.e., a level or dose below which no adverse or toxic effects will occur. Carcinogens, as evaluated by USEPA dose-response methods, are assumed to have no such threshold.

The toxicity or carcinogenicity of a given potential carcinogen is generally described by a slope factor (SF), usually given in units of (mg/kg-day)⁻¹. Slope factors are derived for chronic or lifetime exposures. The greater the value of the slope factor, the more likely the USEPA believes that a given concentration of a chemical may result in cancer. The cancer slope factor is usually obtained from animal studies, and is the slope of a line generated from a probability distribution of dose-response experiments using conservative models and assumptions.

In assessing the carcinogenic potential of a constituent, the USEPA classifies the constituent into one of the following groups, according to the "weight of evidence" from epidemiological studies and/or animal studies:

Group A - Human Carcinogen (sufficient evidence of carcinogenicity in humans);

Group B - Probable Human Carcinogen (B1--limited evidence of carcinogenicity in humans; B2--sufficient evidence of carcinogenicity in animals with inadequate or lack of evidence in humans);

Group C - Possible Human Carcinogen (limited evidence of carcinogenicity in animals and inadequate or lack of human data);

Group D - Not Classifiable as to Human Carcinogenicity (inadequate or no evidence);

Group E - Evidence of Noncarcinogenicity for Humans (no evidence of carcinogenicity in adequate studies).

Quantitative carcinogenic risk assessments are generally performed for chemicals in Groups A and B, and on a case-by-case basis for chemicals for Group C.

Quantitative carcinogenic risk assessments are not performed for chemicals in Groups D or E (USEPA, 1989b).

Another quantitative form of carcinogenic potential occasionally given instead of a slope factor is a "unit cancer risk" (UCR) value. The unit cancer risk is route-specific (i.e., air or water), and is expressed as the amount of risk associated with a given constituent per concentration unit in air or water (e.g., risk per $\mu\text{g/L}$ of water).

A slope factor may be approximated from the unit risk values by using standard intake assumptions (e.g., ingestion of 2 liters of water/day) and solving for the slope factor in the following equation:

Risk per $\mu\text{g/L}$ (water) =

$$\text{Slope Factor per } 1/(\text{mg/kg/day}) \times 1/70 \text{ kg} \times 2 \text{ liters/day} \times 10^{-3} \text{ mg}/\mu\text{g}$$

As further discussed in Section 2.6.3, in the risk assessment, potential health risks from exposure to constituents are estimated by using these toxicity values, along with measured concentrations of the site constituents in relevant site media. The measured concentrations of the constituents are used with various intake factors (e.g., rate of water ingestion) and the toxicity values to estimate potential human health risks. For carcinogens, the intake values are then multiplied by the appropriate slope factors to estimate the potential frequency of cancer risks (e.g., 1 in 100,000 or 1×10^{-5} risk). For noncarcinogens, the intake values are expressed as a ratio with the appropriate RfD value.

The USEPA has values available only for a selected list of hazardous chemicals available in IRIS and is constantly updating the list and the values. Consistent with USEPA's risk assessment guidance (USEPA, 1989b), for the purpose of quantifying potential risks posed by ingestion of shallow groundwater, if RfD or SF values were not available (or able to be derived) either from IRIS (as of July 15, 1991) or the latest HEAST (USEPA, 1991b), potential risks will not be quantified in the Risk Characterization (Section 2.6.3).

2.6.2.1 Available Oral Toxicity Values for Potential Carcinogens. A summary of the latest available oral slope factors for constituents in shallow

groundwater at the RMI Sodium Plant site which have potential carcinogenic effects is presented in Table 2-29. Complete toxicity profiles for each of the potential carcinogens which were on file in IRIS (July 15, 1991) are given in Appendix B. As noted in Table 2-29, for each potential carcinogen of interest, the following information is presented: the latest available slope factor; the weight of evidence; the noted critical health effects; the type of study which was the basis for the development of the oral SF; and the source of information.

Oral slope factors were available for only two of the 13 constituents evaluated: As and Pb. The slope factor for As (an "A" class carcinogen) was calculated from the proposed UCR value of 5×10^{-5} ug/L (IRIS, as of July 15, 1991) using the equation presented above. A slope factor for Pb (a "B2" class carcinogen) was not available in either IRIS (as of July 15, 1991) or the latest HEAST (USEPA, 1991b), but the USEPA Region III recommends an oral slope factor of 0.04 mg/kg-day be used to quantify potential exposures to lead (Personal Communication, 1990). Of the remaining 11 constituents, six are listed as having a "D" weight of evidence class (i.e., inadequate data or no evidence of human carcinogenicity): Cu, cyanide, Hg, Se, Ag, and Zn. As noted earlier, quantitative risk assessments are usually not performed for carcinogenic class D. For the remaining five constituents (Ba, Cd, Cr (both the (III) and (VI) valence states were evaluated), and Ni), current oral slope factors were not available in either IRIS (as of July 15, 1991) or the latest HEAST (USEPA, 1991b) and these constituents have not been classified as to weight of evidence.

2.6.2.2 Available Oral Toxicity Values for Noncarcinogens. The format of toxicity information for the noncarcinogens of interest given in Table 2-30 is similar to that previously presented for the potential carcinogenic values. Complete toxicity profiles for each noncarcinogen which were on file in IRIS (July 15, 1991) are given in Appendix B. Instead of oral slope factors, oral reference dose factors are given; the "weight of evidence" is replaced by an uncertainty factor (UF) indicating the confidence of the USEPA in the RfD value presented.

Of the 13 constituents evaluated, all but Pb and Cu had an RfD value that was listed in IRIS (as of July 15, 1991) or the latest HEAST (USEPA, 1991b). USEPA Region VI recommends that an RfD value for Cu be calculated using the maximum contaminant level (MCL) of 1.3 mg/L as follows (Personal Communication, 1991):

TABLE 2-29

ORAL TOXICITY VALUES FOR POTENTIAL CARCINOGENS OF INTEREST

Constituent	CAS No.	Oral Slope Factor (SF) (mg/kg-day) ⁻¹	Weight of Evidence Classification ^a	Critical Effect(s)	SF Basis (Animal; Exposure)	Source	Remarks
Arsenic	7440-38-2	1.8 ^b	A	Skin cancer, lung cancer	Human; occupational (dietary)	HEAST (1/91); IRIS (7/15/91) ^d	Calculated from proposed 5E-05 µg/l unit risk.
Barium	7440-39-3	NA ^c	NA	NA	NA	--	
Cadmium	7440-43-9	NA	NA	NA	NA	--	
Chromium(III)	16065-83-1	NA	NA	NA	NA	--	
Chromium(VI)	7440-47-3	NA	NA	NA	NA	--	
Copper	7440-50-8	NA	D	Inadequate evidence of carcinogenic properties	NA	IRIS (7/15/91)	SFs not available for Group D constituents.
Cyanide	57-12-5	NA	D	Inadequate evidence of carcinogenic properties	NA	IRIS (7/15/91)	SFs not available for Group D constituents.
Lead	7439-92-1	0.04 ^e	B2	Renal tumors, affects gene expression	Ret, mouse; oral (diet), subcutaneous	IRIS (7/15/91)	Critical effects and SF basis given in IRIS (7/15/91). SF not available in IRIS (7/15/91) or HEAST (1/91).
Mercury	7439-97-6	NA	D	Inadequate evidence of carcinogenic properties	NA	IRIS (7/15/91)	SFs not available for Group D constituents.
Nickel	7440-02-0	NA	NA	NA	NA	--	
Selenium	7782-49-2	NA	D	Inadequate evidence of carcinogenic properties	NA	IRIS (7/15/91)	SFs not available for Group D constituents.
Silver	7440-22-4	NA	D	Inadequate evidence of carcinogenic properties	NA	IRIS (7/15/91)	SFs not available for Group D constituents.
Zinc	7440-66-6	NA	D	Inadequate evidence of carcinogenic properties	NA	IRIS (7/15/91)	SFs not available for Group D constituents.

^a A = Human carcinogen; B1/B2 = probable human carcinogen; C = possible human carcinogen; D = not classifiable as to human toxicity.

^b Slope factor calculated from a unit risk of 5E-05 µg/l that has been proposed by the Risk Assessment Forum and is scheduled for Science Advisory Board review.

^c NA = Not available in IRIS as of date shown or latest HEAST (1/91). Complete toxicity profiles for each constituent available from IRIS are given in Appendix 2. See text for detailed explanation.

^d Dashes (--) indicate that toxicity information was not available from either IRIS (7/15/91) or HEAST (1/91).

^e Slope factor for lead recommended by USEPA Region III; (Personal Communication, 1990).

$$\begin{aligned}\text{Oral RfD} &= (1.3 \text{ mg/liter})(2 \text{ liters/day})(1/70 \text{ kg}) \\ &= 0.037 \text{ mg/kg-day}\end{aligned}$$

Cr has two values listed which are related to its valence state--one for the (VI) valence state (0.005 mg/kg-day for Cr (VI)), and a much higher value for the (III) valence state (1.0 mg/kg-day for Cr (III)), which is the less toxic form of Cr. Because only total Cr was measured in shallow groundwater samples, it is unknown which form of Cr is present, and as discussed further in Section 2.6.3, as a conservative measure, both forms of Cr will be utilized in the risk characterization.

As shown in Table 2-30, the available (or calculated RfD) values ranged from 0.0003 mg/kg-day for Hg to 1.0 mg/kg-day for Cr (III). Uncertainty factors ranged from 1.0 for As to 1,000 for Hg. A variety of critical noncarcinogenic effects are noted in Table 2-30, including kidney effects, anemia, and weight loss (Table 2-31 contains a glossary of toxicity terms which are useful in evaluating the types of critical effects listed in this table). As noted in the "RfD Basis" column in Table 2-30, the available RfD values are based on human and rat toxicity studies. Constituents which have toxicity studies which are based on humans are As, Ba, Cd, Cu, Se, Ag, and Zn.

2.6.3 Risk Characterization

This section describes the final step of the human health risk assessment process, the Risk Characterization. In this section, the shallow groundwater intakes previously presented in Section 2.6.1 and the toxicity assessment information previously given in Section 2.6.2 are integrated into quantitative expressions of potential risk. The methodology from the current federal risk assessment guidance (USEPA, 1989b) was used for characterizing risk, and in the following sections, this methodology is described. There are separate discussions for potential carcinogenic and noncarcinogenic effects because the methodology differs for these two modes of chemical toxicity.

For potential human carcinogens, risks are estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to the

TABLE 2-30

ORAL TOXICITY VALUES FOR POTENTIAL NONCARCINOGENS OF INTEREST

Constituent	CAS No.	Oral Chronic RfD (mg/kg-day)	Confidence in RfD Uncertainty Factor (UF)	Critical Effect(s)	RfD Basis (Animal; Exposure)	Source	Remarks
Arsenic	7440-38-2	1.0E-03 ^a	UF=1	Keratosis and hyperpigmentation	Human; oral (dietary)	HEAST (1/91)	Under review by RfD Work Group; RfD value not available in IRIS (7/15/91).
Barium	7440-39-3	0.05 ^b	Medium; UF=3	Increased blood pressure	Human; oral (drinking water)	HEAST (1/91)	RfD value not available in IRIS (7/15/91).
Cadmium	7440-43-9	5.0E-04	High; UF=10	Renal damage	Human; oral (water or food)	IRIS (7/15/91)	RfD value for water.
Chromium(III)	16065-83-1	1.0	Low; UF=100	No effect observed	Rat; oral (dietary)	IRIS (7/15/91)	
Chromium(VI)	7440-47-3	5.0E-03	Low; UF=500	No effects reported	Rat; oral (drinking water)	IRIS (7/15/91)	
Copper	7440-50-8	0.037 ^c	NA ^d	Local gastrointestinal irritation	Human; NA	HEAST (1/91)	Inadequate data for RfD. RfD value not available in IRIS (7/15/91).
Cyanide	57-12-5	0.02	Medium; UF=100	Weight loss, thyroid effects and myelin degeneration	Rat; oral (dietary)	IRIS (7/15/91)	
Lead	7439-92-1	NA	NA	NA	NA	-- ^e	
Mercury	7439-97-6	3.0E-04	UF=1000	Kidney effects	Rat; oral	HEAST (1/91)	RfD value not available in IRIS (7/15/91).
Nickel	7440-02-0	0.02	Medium; UF=100	Reduced body and organ weight	Rat; oral (dietary)	IRIS (7/15/91)	
Selenium	7782-49-2	5.0E-03	Medium; UF=3	Clinical selenosis	Human; oral (dietary)	IRIS (7/15/91)	
Silver	7440-22-4	3.0E-03	Medium; UF=2	Argyria	Human; therapeutic dosage	IRIS (7/15/91)	
Zinc	7440-66-6	0.2	UF=500	Anemia	Human; therapeutic dosage	HEAST (1/91)	Under review by RfD Work Group; RfD value not available in IRIS (7/15/91).

^a Scientific notation is equivalent to multiplying by 10 to an exponent, e.g., 1.0E-03 is equivalent to 1.0×10^{-3} .

^b IRIS has an RfD of 0.07 mg/kg-day as of 7/15/91. HEAST (1/91) has an RfD of 0.05 mg/kg-day that will replace the value in IRIS.

^c The RfD value for copper was calculated from maximum contaminant level (MCL) of 1.3 mg/l as described in text.

^d NA - Not available in IRIS as of date shown or latest HEAST (1/91). Complete toxicity profiles for each constituent available from IRIS are given in Appendix 2. See text for detailed explanation.

^e Dashes (--) indicate that toxicity information was not available from either IRIS or HEAST.

TABLE 2-31

GLOSSARY OF RELEVANT TOXICITY TERMS^a

anemia	- a condition marked by significant decreases in hemoglobin concentration and in the number of circulating red blood cells.
argyria	- a dusky-gray or bluish discoloration of the skin and mucous membranes produced by the prolonged administration or application of silver preparations.
gastrointestinal	- pertaining to that portion of the digestive tract including the stomach, intestine, and all accessory organs.
hyperpigmentation	- increased pigmentation.
keratosis	- any disease of the skin characterized by an overgrowth of the cornified epithelium.
myelin	- a soft, white fatty substance that forms a sheath around certain nerve fibers.
renal	- pertaining to the kidney.
selenosis	- selenium poisoning.
subcutaneous	- pertaining to the tissues beneath the outer layer of skin.
thyroid	- an endocrine gland found in all vertebrates that produces, stores, and secretes the thyroid hormones.

^aSource: Parker, S.P., (ed.), 1989.

potential carcinogen(s), i.e., incremental or excess individual lifetime cancer risk (e.g., 1×10^{-5} or 1 in 100,000).

Slope factors (SF) were discussed in detail and presented in Section 2.6.2. The SF for a given constituent converts estimated chronic daily intakes (CDIs, previously discussed in Section 2.6.1) averaged over a lifetime of exposure to the potential incremental risk of an individual developing cancer in a lifetime. Because relatively low intakes (compared to those experienced by test animals) are expected to result from environmental exposures, it is generally assumed that the dose-response relationship will be linear in the low-dose portion of the multistage model dose-response curve (USEPA, 1989b). Under this assumption, the slope factor is a constant, and risk will be directly related (i.e., proportional) to intake. Thus, the linear form of the carcinogenic risk equation given below is used for chemical-specific risks:

$$\text{Chemical-Specific Carcinogenic Risk} = \text{CDI} \times \text{SF}$$

where:

Chemical-Specific Carcinogenic Risk = a unitless probability (e.g., 2×10^{-5}) of an individual developing cancer;

CDI = chronic daily intake averaged over 70 years (mg/kg-day); and

SF = slope factor, expressed in (mg/kg-day)⁻¹

Because the slope factor is usually an upper 95th percent confidence limit of the probability of response based on experimental animal data used in the multistage model, the carcinogenic risk estimate is an upper-bound estimate. This means that the "true" risk will most likely not exceed the risk estimate derived through use of this model, and will most likely be less than that predicted with the model.

The measure used to describe the potential for noncarcinogenic toxicity to occur in an individual is not expressed as the probability of an individual encountering an adverse effect. At the present time, the USEPA does not use a probabilistic approach to estimating the potential for noncarcinogenic health effects. Instead, the

potential for noncarcinogenic effects is evaluated by comparing an exposure level (i.e., the CDI) over a specific time period with a reference dose value (RfD). This ratio of exposure to toxicity is called a "hazard quotient", and the formula is given below (USEPA, 1989b):

$$\text{Chemical-Specific Noncarcinogenic Hazard Quotient} = \text{CDI/RfD}$$

where:

CDI = exposure level (i.e., chronic daily intake), in mg/kg-day; and

RfD = reference dose, in mg/kg-day

The noncarcinogenic hazard quotient assumes that the level of exposure associated with the RfD is below that which is associated with adverse health effects (including sensitive populations). If the CDI exceeds this threshold (i.e., if CDI/RfD exceeds unity), there may be a potential for noncarcinogenic effects. As a rule, the greater the value of CDI/RfD above unity, the greater the level of concern. However, the ratios of CDI/RfD are not statistical probabilities; a ratio of 0.001 does not mean that there is a one in one thousand chance of the effect occurring. Further, it is important to emphasize that the level of concern does not increase linearly as the RfD is approached or exceeded, because the values of RfDs do not have equal degrees of certainty and are not based on the same severity of toxic effects. Thus, the slopes of the dose-response curve in excess of the RfD can range widely depending on the substance.

At the RMI Sodium Plant site, several constituents have been identified as being potentially of interest in shallow groundwater. Estimates of risk or hazard potential generated by considering one chemical at a time might underestimate the risk associated with simultaneous exposures to several substances. Therefore, the USEPA recommends assessing the overall potential for carcinogenic and noncarcinogenic effects posed by multiple chemicals simultaneously for a given exposure route. Although the calculation procedures differ for carcinogenic and noncarcinogenic effects, both sets of procedures assume dose additivity, because information on specific mixtures of constituents is rarely available.

The equation for estimating the incremental individual lifetime carcinogenic risks for simultaneous exposure to several carcinogens (USEPA, 1989b) is given below:

$$\text{Total Carcinogenic Risk} = \Sigma \text{Chemical-Specific Risk}_i$$

where:

Total Risk = the total carcinogenic risk, expressed as a unitless probability;

Chemical-Specific Risk_i = the risk estimate for the ith substance

The use of this method assumes independence of action by the compounds involved; in other words, that there are no synergistic or antagonistic chemical interactions, and that all chemicals produce the same effect (i.e., cancer).

To assess the overall potential for noncarcinogenic effects posed by more than one chemical, a hazard index (HI) approach has been developed. This approach assumes that simultaneous subthreshold exposures to several chemicals could result in an adverse health effect. It also assumes that the magnitude of the adverse effect will be proportional to the sum of ratios of the exposures to acceptable exposures. The hazard index is equal to the sum of the chemical-specific hazard quotients per exposure pathway. When the hazard index exceeds unity (1.0), there may be concern for potential health effects. While any single chemical with an exposure level greater than the toxicity value will cause the hazard index to exceed unity, for multiple chemical exposures, the hazard index can also exceed unity even if no single chemical exposure exceeds its RfD. The equation for calculating the noncarcinogenic hazard risk is given below:

$$\text{Noncarcinogenic Hazard Index} = E_1/\text{RfD}_1 + E_2/\text{RfD}_2 + \dots + E_i/\text{RfD}_i$$

where:

E_i = exposure level (or CDI) for the ith substance;

RfD_i = reference dose for the ith substance

By using the chronic daily intakes estimated in Section 2.6.1, the toxicity factors from Section 2.6.2, and the methods described above, chemical-specific and total